Land Use in LCA (Subject Editor: Llorenç Milà i Canals)

Assessment of Land Use Impacts on the Natural Environment

Part 2: Generic Characterization Factors for Local Species Diversity in Central Europe

Thomas Koellner* and Roland W. Scholz

Swiss Federal Institute of Technology, Department of Environmental Sciences, Natural and Social Science Interface (ETH-NSSI), ETH-Zentrum CHN, 8092 Zurich, Switzerland

* **Corresponding author** (thomas.koellner@env.ethz.ch)

Part 1: An analytical framework for pure land occupation and land use change [Int J LCA 12 (1) 16–23 (2007)]
Part 2: Generic characterization factors for local species diversity in Central Europe [Int J LCA 13 (1) 32–48 (2008)]

Preamble. This series of two papers is based on a PhD thesis (Koellner 2003) and develops a method how to assess land use impacts on biodiversity in the framework of LCA. **Part 2** rests on a much richer database compared to the thesis in order to quantify generic characterization factors for local species' richness. **Part 1** further expands the analytical framework of the thesis for pure land occupation and land use change.

DOI: http://dx.doi.org/10.1065/lca2006.12.292.2

Please cite this paper as: Koellner T, Scholz RW (2008): Assessment of Land Use Impacts on the Natural Environment. Part 2: Generic Characterization Factors for Local Species Diversity in Central Europe. Int J LCA 13 (1) 32–48

Abstract

Goal, Scope and Background. Land use is an economic activity that generates large benefits for human society. One side effect, however, is that it has caused many environmental problems throughout history and still does today. Biodiversity, in particular, has been negatively influenced by intensive agriculture, forestry and the increase in urban areas and infrastructure. Integrated assessment such as Life Cycle Assessment (LCA), thus, incorporate impacts on biodiversity. The main objective of this paper is to develop generic characterization factors for land use types using empirical information on species diversity from Central Europe, which can be used in the assessment method developed in the first part of this series of paper.

Methods. Based on an extensive meta-analysis, with information about species diversity on 5581 sample plots, we calculated characterization factors for 53 land use types and six intensity classes. The typology is based on the CORINE Plus classification. We took information on the standardized α-diversity of plants, moss and mollusks into account. In addition, threatened plants were considered. Linear and nonlinear models were used for the calculation of damage potentials (EDP^S). In our approach, we use the current mean species number in the region as a reference, because this determines whether specific land use types hold more or less species diversity per area. The damage potential calculated here is endpoint oriented. The corresponding characterization factors EDP^S can be used in the Life Cycle Impact Assessment as weighting factors for different types of land occupation and land use change as described in Part 1 of this paper series.

Results. The result from ranking the intensity classes based on the mean plant species number is as expected. High intensive forestry and agriculture exhibit the lowest species richness (5.7–5.8 plant species/m^2), artificial surfaces, low intensity forestry and non-use have medium species richness (9.4–11.1 plant species/m^2) and low-intensity agriculture has the highest species richness (16.6 plant species/m^2). The mean and median are very close, indicating that the skewedness of the distribution is low. Standard error is low and is similar for all intensity classes. Linear transformations of the relative species numbers are linearly transformed into ecosystem damage potentials (EDP^S_{linear}). The integration of threatened plant species diversity into a more differentiated damage function $EDP^{S_{total}}_{linear}$ makes it possible to differentiate between land use types that have similar total species numbers, but intensities of land use that are clearly different (e.g., artificial meadow and broad-leafed forest). Negative impact values indicate that land use types hold more species per m^2 than the reference does. In terms of species diversity, these land use types are superior (e.g. near-to-nature meadow, hedgerows, agricultural fallow).

Discussion. Land use has severe impacts on the environment. The ecosystem damage potential EDP^S is based on assessment of impacts of land use on species diversity. We clearly base EDP^S factors on α-diversity, which correlates with the local aspect of species diversity of land use types. Based on an extensive meta-analysis of biologists' field research, we were able to include data on the diversity of plant species, threatened plant species, moss and mollusks in the EDP^S. The integration of other animal species groups (e.g. insects, birds, mammals, amphibians) with their specific habitat preferences could change the characterization factors values specific for each land use type. Those mobile species groups support ecosystem functions, because they provide functional links between habitats in the landscape.

Conclusions. The use of generic characterization factors in Life Cycle Impact Assessment of land use, which we have developed, can improve the basis for decision-making in industry and other organizations. It can best be applied for marginal land use decisions. However, if the goal and scope of an LCA requires it this generic assessment can be complemented with a site-dependent assessment.

Recommendations and Perspectives. We recommend utilizing the developed characterization factors for land use in Central Europe and as a reference methodology for other regions. In order to assess the impacts of land use in other regions it would be necessary to sample empirical data on species diversity and to develop region specific characterization factors on a worldwide basis in LCA. This is because species diversity and the impact of land use on it can very much differ from region to region.

Keywords: Generic assessment; impacts; land use; LCA; species diversity

Introduction

Land use is an economic activity that causes many environmental problems. Consequently, *land use* has been introduced in LCA as impact category (Heijungs et al. 1997, Udo de Haes et al. 1999). In particular biodiversity has been negatively influenced by intensive agriculture, forestry and the increase of urban areas and infrastructure. The measurement of land use impacts on biodiversity, however, is a complex task, because a widely accepted definition of biodiversity does not exist. In LCA, some indicators were proposed for *species diversity* and *ecological diversity*. Indicators for species diversity include number/percentage of vascular plant species (Koellner 2000, Koellner 2003, Vogtländer et al. 2004), number/percentage of threatened vascular plant species (e.g. Goedkoop and Spriensma 1999, Koellner 2003, Müller-Wenk 1998), species accumulation rate (Koellner 2000, Lindeijer 2000) or probability of species occurrence (Wiertz van Dijk and Latour 1992). The following indicators were proposed for ecological diversity: Structural diversity of forest habitats (Giegrich and Sturm 1996) and area/percentage of rare ecosystems (Müller-Wenk 1998, Schenck 2001, Vogtländer et al. 2004). The usefulness of those approaches for decision-makers in industry and administration very much depends on the availability of basic ecological information from environmental sciences. At the same time the information provided must be functional and meaningful to decision-makers (Werner and Scholz 2002). We propose to develop a set of generic characterization factors, which express the potential damage for ecosystems or more specific for species diversity, being an important aspect of ecosystems. The main goal of this paper is to perform a meta-analysis on land use and species diversity and to propose a method for the assessment of land use impacts on species diversity on the local scale, which is consistent with the framework of LCA. Accordingly, the method must be generic and is generally not site-dependent. Udo de Haes (2006) proposes to implement such generic weighting schemes of species assemblages of different types of land use in the framework of LCA. To address site-dependent impacts of land use other approaches like environmental impact assessment (EIA) are more appropriate. In order to provide decision-makers with ecological information, we quantify the potential impact of 53 land use types (ranging from continuous urban to near-to-nature forestry) based on reliable, published data. We expand upon an existing meta-analysis for local species diversity (Koellner 2003) and calculate characterization factors on the basis of empirical data from 5,581 plots. In the meta-analysis we differentiate between all plant species and threatened plant species, as well as moss and mollusks. Specific problems posed by a meta-analysis of species richness and land use types are addressed.

1 Method for Developing Characterization Factors for Species Diversity

1.1 Endpoint definition and indicators of species diversity

For the quantification of land use impacts on species diversity in LCA it is essential to clearly define assessment endpoints for the calculation of characterization factors labeled *Ecosystem Damage Potential* with respect to species diversity or in short EDP^S. For operational integration of biodiversity into LCA we proposed species diversity for endpoint definition (Koellner 2000, Koellner 2003). Genetic diversity is not operational, because there is a severe lack of information about the impact of land use activities on the genetic diversity of populations and species. Ecological diversity is included indirectly, because in the chain of cause and effect, the homogenization of habitats (i.e., reduction of ecological diversity) is an intermediate factor that directly contributes to the reduction of species diversity.

Endpoints for species diversity should be defined on different scales. It is essential to distinguish between α-, β- and γ-diversity (in sensu MacArthur 1965, Whittaker 1972, Whittaker et al. 2001), because underlying ecological processes interplay on multiple scales (Levin 2000). The mean species diversity for a single land use type can be referred to as α-diversity. It is defined on the local scale for homogenous land use types (e.g. mean species number of 1 m^2 plantations versus 1 m^2 near-to-nature forests). β-Diversity is defined from local to regional scales. In this paper it stands for the species turnover between sample plots of one land use type (Whittaker 1972, Whittaker et al. 2001). Generally, the value for β-diversity measures the species diversity between sample plots. It is high when sample plots differ with respect to their species community. It is low when the species composition of sample plots is very similar. It increases as the diversity of habitats and, hence, the environmental heterogeneity increases (Alard and Podevigne 2000, Balvanera et al. 2002, Whittaker 1972). In Switzerland, for example, β-diversity is large for pioneer and weed species and small for fertilized meadow species; this clearly reflects the degree of homogeneity of respective habitats (Koellner et al. 2004). Finally, the species diversity of an entire region is defined as γ-diversity and is a function of β-diversities at intraregional scales (Balvanera et al. 2002, Whittaker 1972).

1.2 Development of characterization factors for species diversity

Quantifying species diversity is challenging because of difficulties in measuring species abundance and distribution (Magurran 1996). In experimental settings (e.g. Hector et al. 1999) and in landscape ecology (e.g. Wohlgemuth 1998) species richness (i.e., the number of species per sample plot) was used as a proxy for diversity. To overcome the problem of abundance measurement a probabilistic method has been used for estimating species diversity (Hurlbert 1971, Palmer 1990, Simberloff 1978). Based on the presence or absence of data on the species in the sample plots, they calculated the expected number of species using a rarefaction function. The discontinuous rarefaction function integrates data on the species' commonness or rarity in a given region. However, data requirements for rarefaction functions are less demanding than for indices like the Shannon-Wiener Index (Shannon 1948) and the Simpson Index (Simpson 1949), both of which require data on abundance.

For reliable characterization factors, the nonlinearity of the relationship between area and species number should be taken into account. Models used for fitting species-area

samples portray a monotonically increasing curve, which is steep at the beginning and gradually becomes flat (He and Legendre 1996). That is to say that, the first deviation of the functions is decreasing. Three models are commonly used for such curves: the power model (Arrhenius 1921), the exponential model (Gleason 1922, Gleason 1925), and the logistic model (Archibald 1949). We used the widespread power (log-log) model according to Arrhenius (1921)

$$S = cA^z \tag{1}$$

where S is the species number; A the area of the plot; parameter c (measure for species richness) and parameter z (measure for species accumulation rate). The transformed power model

$$\ln S = \ln c + z \ln A \tag{2}$$

shown in equation [2] has two parameters. The parameter ln c (y-intercept) indicates the species richness of a sample standardized for $A = 1$. The parameter z (slope) denotes species accumulation rates and was proposed as a measure for β-diversity (Koellner et al. 2004, Ricotta et al. 2002).

1.3 Data sources and calculation of α-diversity

The goal is to derive generic characterization factors for all types of land use. It was not possible to gather data on all species groups. Therefore, we have chosen the vascular plant species as a proxy for the total species richness. One reason for this choice is the existence of reliable data for a wide variety of land use types. In addition, vascular plant species constitute terrestrial ecosystems and its diversity correlates highly with other species groups' diversity (Duelli and Obrist 1998). To check for correlations, the numbers of moss and mollusk species were assessed, based on the plots from the Biodiversity Monitoring Switzerland (BDM 2004).

In order to develop characterization factors for plant species richness of individual land use types we performed a meta-analysis of published investigations, most of them from vegetation science. Plant species and composition of the vegetation types were investigated according to the methods from Braun-Blanquet. The distribution of samples of land use types and their sources are given in **Table 1**.

In order to standardize the species number S we used one single species-area relationship for all land use types. We adjusted the species-area relationship according to equation [2], which yields a straight regression line on a ln-ln scale fitting all empirical data. Next we calculated the standardized species number S_{lm^2} by shifting the data points, in parallel with the regression line, to the standard area size. This eliminates the aspect of S that is attributable to area size. The standardized species number S_{lm^2} for 1 m² is calculated as

$$S_{\mathrm{lm}^2} = S_{plot} - \Delta S \tag{3}$$

where S_{plot} is the species number measured on a plot of size A_{plot} in the field (which varies between different empirical studies used here) and ΔS is that part of the species number which can be attributed to the area rather than the land use types. It is calculated as

$$\Delta S = S'_{plot} - S'_{1\mathrm{m}^2} = cA_{plot}z - cA_{1\mathrm{m}^2}z \tag{4}$$

where S'_{plot} is the average species number for the area A_{plot} and $S'_{1\mathrm{m}^2}$ is the average species number for the standard area $A_{1\mathrm{m}^2}$. The average species number takes all land use types into consideration and is calculated using the regression line (see **Fig. 2**). The species number standardized for 1 m² was calculated as:

$$S_{1\mathrm{m}^2} = S_{plot} - c(A_{plot}z - A_{1\mathrm{m}^2}z) \tag{5}$$

For each species group we calculated mean species number standardized for 1 m², standard error of mean (calculated as $\sigma / \sqrt{n}$), median, minimum and maximum of species number. In order to compare the different species groups, we performed a correlation analysis on the number of species per group (plants, threatened plants, moss, and mollusks) and determined which correlations are significant.

The number of threatened species found in BDM was also taken into account. The reason for this is that the indicator average species number would underestimate the ecological value of ecosystem types, which carry few but species of any threat status. In Switzerland for example 31.5% of the 3,144 vascular plant species are extinct, endangered or vulnerable (BUWAL 2002, IUCN 2001), 13.6% are near threatened, and 48.8% of the species are not threatened (i.e. they are of the least concern in IUCN terms). Obviously the occurrence of those species should be weighted more. The necessary data for that were only available for land use plots investigated in the BDM (2004).

1.4 Data source and calculation of β-diversity

In order to calculate β-diversity we used the rarefaction function (Koellner et al. 2004). This method allows species-area relationships to be constructed out of species lists per sample plot. The resulting curve is discontinuous since the calculation is based on the hyper-geometrical distribution. The method gives the expected number of species if n out of N sample plots are randomly chosen. For comparison of the slope of the curves for different land use types the continuous power function was fitted to the rarefaction function.

In order to calculate the rarefaction curves, a consistent set of data is needed. For each sample plot, both the species number and the complete species list must be known. In addition, a large number of sample plots are needed to assess the slopes reliably. Data were taken from the Biodiversity Monitoring Switzerland (BDM 2004). For only six out of 33 land use types 39 or more sample plots for each land use type were available. Since the species turnover was expected to be different for areas < 800 m above sea level and mountainous areas > 800 m data sets were split into these two groups. Some land use types (e.g. bare rock) occur in Switzerland only > 800 m.

Table 1: Number of sample plots per source and land use type

CORINE Plus ID	(Adam 1995)	(Albracht 1997)	(BDM 2004)	(Bigler et al. 1998)	(Bruelheide 1995)	(Callauch 1981)	(Döring-Mederake 1991)	(Ewald 1997)	(Flückiger 1999)	(Grüttner 1990)	(Kisteneich 1993)	(Lips et al. 1997)	(Manz 1997)	(Murmann-Kristen 1987)	(Reidl 1989)	(Schreiber 1995)	(Schulte 1985)	Schwab, unpublished	(Sukopp 1990)	(Wittwer et al. 1997)	(Wohlgemuth 1992)	(Zerbe 1999)	(von Oheimb 2003)	Total
111			5												10		50							65
112			32												59		61							152
113															24				3					27
114																	17							17
121			5												29		24							58
122			18												41		32		3					94
125															26									26
132			3																					3
134															10									10
141			1												75		35							111
142		16	2																					18
211			82	103		88						148				46		120						587
221			4	48																				52
222			9																					9
231			168	214	462					100		63								61				1068
244			2																					2
245				11																				11
311	10		57				501				122			101							223	97	263	1374
312	15		87					72						40								49		263
313	27		92					172						63								56		410
314				51					27															78
321			89		78					120		4	40											331
322			30										6											36
324			16	44																				60
331			2																					2
332			42																					42
333			31																					31
411			5																					5
412			1		28					605														634
511			5																					5
Total	52	16	788	471	568	88	501	244	27	825	122	215	46	204	274	46	219	120	6	61	223	202	263	5581

1.5 Indicator for γ-diversity

γ-diversity refers to the total number of species in a given region. Land use types, which carry threatened species reduce the probability that species become extinct and thus total number of species decreases in this region. In this sense the indictor average threatened species number on the local scale refers also to the γ-diversity. However, in this paper this indicator could not be calculated due to limited availability of data, which are consistent with those for α- and β-diversity.

1.6 Conversion of effects into damage/benefits

Based on the empirical information on species diversity for specific land use types we develop the characterization factor ecosystem damage potential (EDP^S). In literature we found absolute species numbers, however, these are less meaningful than relative species numbers where a comparison with a reference is made. Species richness on a biogeographical scale varies remarkably. If one divides Europe into 4 diversity zones, the number of vascular plant species per 10,000 km² ranges from 200 to 500 in northern Scandinavia

and from 2,000 to 3,000 in the southern parts of Mediterranean countries (Barthlott 1998, p. 36). Obviously, the impact of occupying a plot of land should be assessed relative to the region where the occupation takes place. Along a similar vein of thought, Lindeijer proposed a map for reference states of plant diversity (2000). To calculate relative species numbers, we chose regional average species richness as a reference for assessing species richness of local plots.

In order to further transform the empirical data on relative species diversity into characterization factors, we considered two options for the effect-damage function: (1) a linear function and (2) a logarithmic one. Both functions are purported to describe the functional relationship between species richness on a plot and ecosystem functions (Schläpfer and Schmid 1999) and are based on theories in ecosystem science (Schulze and Mooney 1994).

Option 1: Linear effect-damage function

The ecosystem damage potential for species diversity EDP^{S}_{linear} can be calculated using a linear relationship as shown in **Fig.** 1b:

$$EDP^{S}_{linear} = 1 - \frac{S_{occ}}{S_{region}} \tag{6}$$

where S_{occ} is the species number of an occupied land use type and S_{region} is the average standardized species number in the region. We took the Swiss Lowland as reference region, which serves a proxy for Barthlott's diversity zone 5. As a consequence land use types with lower species number compared to the reference are treated as detrimental land use types and such with higher species number as beneficial land use types. This calculation is appropriate to account also for threatened species since each species is weighted equally. We calculated $EDP^{S_{total}}_{linear}$ as the unweighted sum of $EDP^{S_{plants}}_{linear}$ and $EDP^{S_{threatened\ plants}}_{linear}$ for each of 5582 local plots. Data on threatened species were only available for a subset of 841 plots and threatened species were only found on 78 of these. The mean and standard error of mean was determined for each type of EDP^{S}_{linear}.

Option 2: Nonlinear effect-damage function

The logarithmic function (**Fig. 1**) is supported by the redundant species hypothesis, that is, that the addition of one species results in a decrease in the marginal growth of utility in terms of ecosystem processes. The logarithmic relationship was taken, because Schläpfer and Schmid (1999) created an expert questionnaire and compiled the results with the finding that this relationship is most likely. Ecosystem processes *EP* are a function of relative species richness (see Fig. 1a)

$$EP = a\left(\ln \frac{S_{occ}}{S_{regional}}\right) + b \tag{7}$$

Based on the previous function the nonlinear function for $EDP^{S}_{nonlinear}$ (see **Fig.** 1b) is calculated as

$$EDP^{S}_{nonlinear} = 1 - EP = 1 - \left\{ a \ln\left(\frac{S_{occ}}{S_{region}}\right) + b \right\} \tag{8}$$

The parameters *a* and *b* in equation [8] were quantified based on the work of Schläpfer and Schmid (1999). Using as 0.27 for *a* and 1 for *b* as parameter estimates, the resulting curve (**Fig.** 1a) has the approximate shape Schläpfer and Schmid had suggested based on the survey of 39 experts. We used the resulting effect-damage curve to transform the relative species number is shown in Fig. 1b.

We calculated $EDP^{S_{total}}_{nonlinear}$ as the unweighted sum of $EDP^{S_{plants}}_{nonlinear}$, $EDP^{S_{moss}}_{nonlinear}$ and $EDP^{S_{mollusks}}_{nonlinear}$ for a consistent set of 841 local plots from the Biodiversity Monitoring Switzerland. Since a regional reference for moss and mollusks is missing, we took S_{ref} of low intensity agriculture instead.

1.7 Classification of land use types

For the typology of land uses we applied the CORINE land cover classification (European Environmental Agency 2000). CORINE includes all the major land cover types in Europe and provides three different levels of classification. Some modifications were necessary, because in their original form, the classifications did not distinguish between low-intensity land use and high-intensity land use. Especially for forestry and agriculture, such distinctions are very important. For example the damage potential for forests, which are close-to-nature, and coniferous plantations is assumed to differ and should be separately assessed. The modified CORINE Plus classification (Koellner 2003) is given in Appendix 1 (see OnlineEdition, DOI: http://dx.doi.org/10.1065/lca2006.12.292.3, pp. 48-1-48-3).

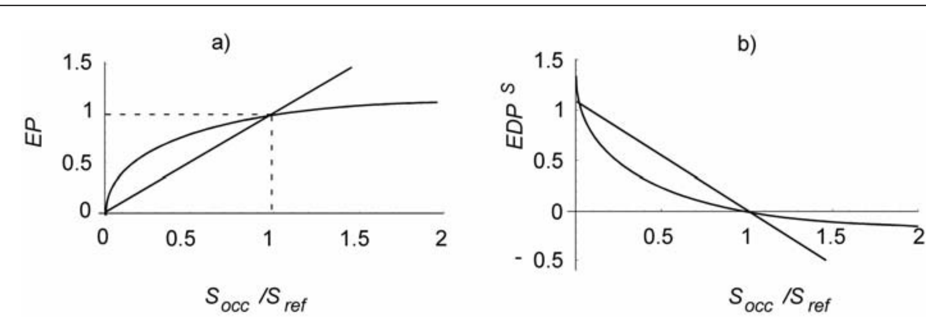

Fig. 1: a) Linear and logarithmic relationships reflecting relative species richness (S_{occ} is the species number on the occupied plot and S_{ref} that of the reference) and ecosystem processes EP. b) The corresponding effect-damage function for ecosystem damage potential EDP^S. The parameters of nonlinear relationship [7] were based on Schläpfer et al. (1999)

2 Results for Characterization Factors of Land Use Types

2.1 Function of species-area relationship used for standardization

The size and species number of all local plots, regional polygons of Switzerland and global biodiversity zones are shown in **Fig. 2.** The standardized species number was calculated based on this graph. Based on the regression function [2], which has a correlation coefficient of R^2 = 0.60, we can calculate $S = 9.58A^{0.21}$ in m². The regression was calculated taking into account all local plots (homogenous land use types) and regional plots from Switzerland (species diversity for mix of land use types based on WSL/FNP without year).

2.2 α-Diversity

The standardized α-diversity for vascular plants, threatened vascular plants, moss and mollusks differ between specific land use types, classes of intensity of land use (**Table 2**). In general, the standardized species number per m² ranges from 3.8 species (sport facilities) to 27.4 species (semi-natural forests with 800 m above sea level). Three general classes of land use types can be distinguished:

- *The species poor land use types* (3.5–8 plant species/m²) include sport facilities, continuous urban area, conventional arable land, intensive meadow, and coniferous plantations. The reason for the low species number, and thus, its low ecological value, is the high-intensity of land use. A mixture of intentional physical and/or chemical measures keeps the species number very low. Due to its low species number bare rock in the Swiss Alps above 800 m belongs also to this group, despite of the fact this natural habitat shows a high number of threatened species.
- Some examples for land use *types with medium species richness* (8–15 plant species/m²) are semi-natural broadleafed forest, organic arable land, and industrial areas with vegetation, green urban, and discontinuous urban area. This group is quite heterogeneous in terms of naturalness and intensity of land use. For example, in artificial meadows (11 plant species/m²), cultivation is very intensive, whereas in broad-leaved forests (10.8 plant species/m²), human impact is less common and intensive. Although the mean number of plant species is almost equal, the forest habitat has more threatened plant species than artificial meadows do.
- Land use types with high species richness (15–28 plant species/m²) are industrial and agricultural fallow, organic meadow, forest edges, agricultural fallow with hedgerows, and natural grassland. This group is also heterogeneous in terms of naturalness and land use intensity. One reason for the high species number is that land is generally not used intensively. Agricultural fallow with hedgerows and forest edges are species rich, because these land use types are heterogeneous in terms of abiotic conditions (e.g., light, moisture). Many land use types with threatened species can be found in this group.

The result from ranking the intensity classes based on the mean plant species number is as expected. High intensive forestry and agriculture exhibit the lowest species richness (5.7–5.8 plant species/m²), artificial surfaces, low intensity forestry and non-use have medium species richness (9.4–11.1 plant species/m²) and low-intensity agriculture has the highest species richness (16.6 plant species/m²). The mean and median are very close, indicating that the skewedness of the distribution is low. Standard error is low and is similar for all intensity classes.

The data for moss and mollusk species is less certain than that for plant species. This is mainly attributable to there being a fewer number of sample plots per land use type, which leads to high standard errors. Those data, however, can be used to make distinctions between land use types, which are similar in plant species number, but very different in other species groups. For example, although the numbers

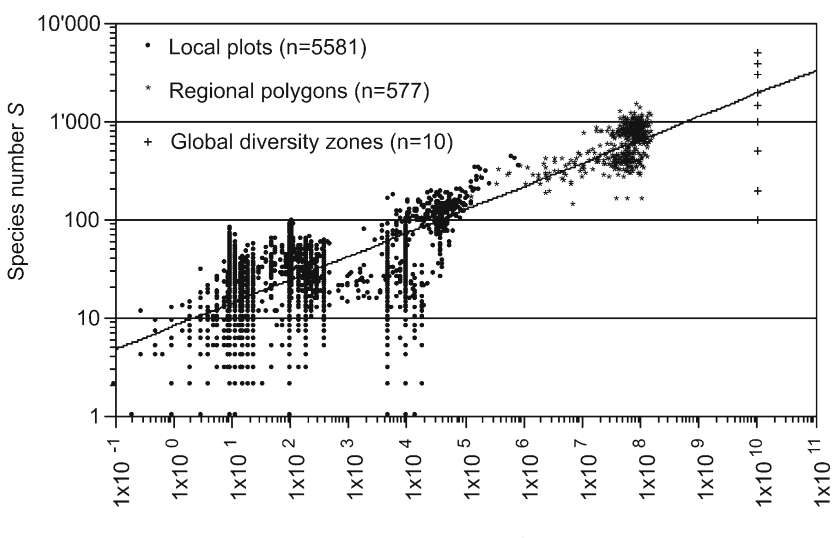

Fig. 2: Regression function (S = 9.58*A^0.21, R² = 0.60) for the area in m² and species number, taking local plots, regional polygons, and global diversity zones into account

Table 2: Standardized species numbers S (vascular plants, moss, mollusks) for specific land use types, intensity classes, and Swiss regions. The area chosen for standardization was 1 m². Calculated for Switzerland on the basis of the BDM data set, there are 1061 vascular species, 519 moss species, and 133 mollusk species

		S_{plants}						$S_{threatened\ plants}$			S_{moss}			$S_{mollusks}$		
CORINE Plus ID		**Mean**	**Std. Error**	**Minimum**	**Median**	**Maximum**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**
	Land-use types															
111	Continuous urban	3.5	0.4	0	3	26	65	0.6	–	1	2.1	0.4	5	7.1	3.6	3
112	Discontinuous urban	9.5	0.4	1	9	32	152	0.6	0.0	3	4.4	0.6	23	4.1	0.5	20
113	Urban fallow	15.5	1.0	8	14	29	27			0			0			0
114	Rural settlement	9.6	0.5	5	10	12	17			0			0			0
121	Industrial units	14.6	6.7	1	16	37	5	0.6	0.0	2	3.3	1.0	4	2.7	1.1	3
121b	Industrial area with vegetation	9.5	0.7	1	9	20	53			0			0			0
122	Road and rail networks	17.7	2.9	1	21	37	18	0.6		1	5.1	0.9	12	5.8	0.9	11
122b	Road embankments	6.4	0.6	1	6	10	13			0			0			0
122d	Rail embankments	14.1	0.7	9	13	28	44			0			0			0
122e	Rail fallow	9.2	0.5	6	9	13	19			0			0			0
125	Industrial fallow	15.7	0.8	8	16	22	26			0			0			0
132	Dump sites	16.8	3.9	11	15	24	3			0	3.1	0.6	3	1	0.4	3
134	Mining fallow	14.8	0.7	11	14	19	10			0			0			0
141	Green urban areas	11.5	0.6	2	12	29	111	0.6	–	1	6.9	–	1	6.2	–	1
142	Sport and leisure facilities	3.8	1.0	1	3	20	18			0	2.8	2.2	2	11.2		1
211	Non-irrigated arable land	9.3	1.6	4	8	21	12	0.6	0.0	2	1.7	0.4	5	1.6	0.4	5
211a	Intensive arable	4.0	0.3	0	4	10	54			0			0			0
211b	Less intensive arable	3.8	0.3	0	3	26	198	0.7	0.1	5	1.1	0.2	15	2.6	0.4	24
211c	Organic arable	10.0	0.5	1	11	15	62			0			0			0
211d	Fibre/energy crops	4.9	0.3	0	5	11	94			0			0			0
211e	Agricultural fallow	16.8	0.5	5	17	32	139	0.6	–	1	4.4	–	1	1.9	–	1
211f	Artificial meadow	11.0	0.5	6	12	16	28			0	1.8	0.2	22	3.3	0.5	23
221	Vineyards	6.7	3.1	2	5	16	4			0	0.6		1	4.8	1.4	4
221b	Organic vineyards	9.1	0.4	5	9	17	48			0			0			0
222	Fruit trees and berry plantations	15.1	2.0	10	16	19	4			0	2.2	1.4	4	4.4	0.5	4
222a	Intensive orchards	13.5	3.1	7	16	17	3	0.6	0.0	2	1.9	0.6	2	3.7	1.3	3
222b	Organic orchards	14	5.3	9	14	19	2			0	0.6		1	4.4	1.9	2
231	Pastures and meadows	15.8	0.7	6	14	35	78			0	2	0.2	60	4.2	0.4	72
	" above 800m	24.7	0.9	10	24	47	86	0.9	0.3	2	5.8	0.6	69	3.3	0.3	70
231a	Intensive pasture and meadows	7.2	0.6	1	7	30	73			0	2.5	0.6	3	3.3	1.3	3
231b	Less intensive pasture and meadows	7.5	0.4	2	6	18	104			0			0			0
231c	Organic pasture and meadows	17.5	0.3	2	17	44	727	0.7	0.1	13			0			0
244	Agro-forestry areas	21.2	3.7	17	21	25	2			0	10.6	5.0	2	2.5	0.6	2
245	Agricultural fallow with hedgerows	20.6	0.7	17	19	25	11			0			0			0
311	Broad-leafed forest	10.8	1.0	1	10	26	31	0.6	0.0	2	7.3	0.7	30	8.1	0.9	29
	" above 800m	9.9	1.2	1	10	26	26			0	7.6	0.9	25	4.7	0.9	21
311a	Broad leafed plantations	7.9	2.0	4	6	16	5			0			0			0
311b	Semi-natural broad-leafed forests	9.3	0.1	1	9	27	1312	0.6	–	1	1.1	0.1	115			0
312	Coniferous forest	6.9	1.1	5	6	10	4			0	9.8	2.9	3	6.2	2.2	3
	" above 800m	13.2	0.8	2	12	29	73			0	10.2	0.6	73	3.7	0.4	66
312a	Coniferous plantations	6.7	0.3	1	6	18	74			0	5.8	0.7	6	3.9	0.4	6
312b	Semi-natural coniferous forests	16.0	0.8	4	14	34	99			0	9.0	0.3	2	4.7	0.9	2
	" above 800m	27.4	1.4	15	29	33	13	0.8	0.1	8			0			0

Table 2: Standardized species numbers S (vascular plants, moss, mollusks) for specific land use types, intensity classes, and Swiss regions. The area chosen for standardization was 1 m². Calculated for Switzerland on the basis of the BDM data set, there are 1061 vascular species, 519 moss species, and 133 mollusk species (*cont'd*)

		S_{plants}						$S_{threatened\ plants}$			S_{moss}			$S_{mollusks}$		
CORINE Plus ID		**Mean**	**Std. Error**	**Minimum**	**Median**	**Maximum**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**
313	Mixed forest	9.9	0.6	1	8	28	83			0	5.7	0.4	35	7.3	0.7	36
	" above 800m	19.3	0.5	3	19	36	223			0	9.0	0.6	46	5.7	0.5	46
313a	Mixed broad-leafed forest	12.6	1.1	9	14	18	8			0			0			0
313b	Mixed coniferous forest	7.1	0.3	5	7	11	35			0	6.9	1.2	2	3.1	0.0	2
313c	Mixed plantations	4.3	0.3	2	4	11	61			0			0			0
314	Forest Edge	18.5	1.1	1	18	40	78			0			0			0
321	Semi-natural grassland	18.3	0.5	2	17	49	331	0.7	0.0	19	9.2	0.6	86	3.3	0.3	66
322	Moors and heath land	14.2	1.1	5	13	29	36	0.9	0.3	2	11.1	0.9	30	3.4	0.8	21
324	Transitional woodland/shrub	17.2	1.0	2	18	34	60	0.6	0.0	2	10.3	1.5	16	4.7	1.0	13
331	Beaches, dunes, and sand plains	5.6	2.5	3	6	8	2			0	1.6	0.9	2	1.9	–	1
332	Bare rock	8.7	0.9	1	7	22	42	1.2		1	6.9	0.6	40	1.9	0.4	18
333	Sparsely vegetated areas	19.8	1.4	5	20	42	31	0.6	0.0	6	11.4	1.0	30	2.4	0.6	23
411	Inland marshes	18.0	3.7	9	16	28	5	1.0	0.2	3	4.1	1.6	5	9.4	2.3	5
412	Peat bogs	7.2	0.2	1	7	24	634			0	3.1	–	1	4.4	–	1
511	Water courses	7.5	2.6	2	7	16	5	0.6		1	4.9	1.2	5	1.7	0.7	3
	Total	11.8	0.1	0	10	49	5581	0.7	0.0	78	6.1	0.2	787	4.2	0.1	617
	Intensity classes															
1	Artificial surfaces	9.4	0.3	0	9	37	481	0.6	0.0	7	3.9	0.4	38	4.3	0.6	31
2	Agriculture high intensity	5.8	0.2	0	5	30	524	0.7	0.1	11	1.6	0.1	47	2.9	0.3	58
2	Agriculture low intensity	16.6	0.2	2	16	49	1214	0.7	0.0	19	9.1	0.6	89	3.3	0.3	70
3	Forestry high intensity	5.7	0.2	1	5	18	140			0	5.8	0.7	6	3.9	0.4	6
3	Forestry low intensity	11.0	0.2	1	9	36	1773			0	3.9	0.3	200	6.3	0.4	86
3	Non-use	11.1	0.2	1	10	42	1120	0.8	0.1	15	9.2	0.5	125	3.4	0.4	83
	Regions in Switzerland															
	Alps	19.7	0.2	12	19	33	202	0.7	0.0	172			0			0
	Jura	17.1	0.4	12	18	23	44	0.7	0.1	33			0			0
	Plateau	15.6	0.3	11	15	24	104	0.1	0.0	23			0			0
	Above timberline	10.7	0.2	3	10	22	215	0.2	0.0	177			0			0
	Lakes	0.5	0.1	0	0	1	28	1.0	0.1	87			0			0
	Worldwide diversity regions															
	DZ5 (Swiss Plateau)			8.9		13.3										
	DZ6 (Swiss Alps)			13.3		17.7										

of plant species in both are similar, broad-leafed forests show higher numbers of moss and mollusk species than artificial meadows do. A correlation analysis was done between plant species, threatened plant species, moss and mollusks species (**Table 3**). The results reveal a relatively clear and highly significant correlation between the number of all plant species and the number of threatened species. The correlation between the number of plant species and the number of moss species is less clear, but still significant.

2.3 β-Diversity

The slope parameter z of the fitted power function was taken as an indicator of β-diversity. The rarefaction curves are based on 39 sample plots for each of six land use types (**Fig. 3**). The fitted curve parameter z for vascular plants reveal no differences between land use types (**Table 4**). For mollusk species, however, β-diversity – and thus, species turnover – is different between land use types. The β-diversity of mollusks is generally low for forests and high for arable land, grassland and bare rock.

2.4 Ecosystem damage potential on the local scale EDP^S

The basis for calculating the local damage is the species number (α-diversity) of a specific land use type S_{occ} standardized for 1 m². β-Diversity was not included in the characterization factor, because not enough sample plots were available for reliably estimating parameters.

Table 3: Correlation of species groups

	S_{plants}	$S_{threatened\ plants}$	S_{moss}	$S_{mollusks}$
S_{plants}				
Pearson Correlation R^2	1.00	0.51**	0.29**	0.02
Probability p (2-tailed)	–	0.00	0.00	0.61
Plot N	6166	570	787	617
$S_{threatened\ plants}$				
Pearson Correlation R^2		1.00	–0.13	–0.05
Probability p (2-tailed)		–	0.29	0.73
Plot N		570	68	62
S_{moss}				
Pearson Correlation R^2			1.00	0.06
Probability p (2-tailed)			–	0.16
Plot N			787	563
$S_{mollusks}$				
Pearson Correlation R^2				1.00
Probability p (2-tailed)				–
Plot N				617

** Correlation is significant at the p=0.01 level (2-tailed)

Linear transformations of the relative species numbers result in ecosystem damage potentials (EDP^{S}_{linear}, **Table 5**). The integration of threatened plant species diversity into $EDP^{S_{total}}_{linear}$ makes it possible to differentiate between land use types that have similar total species numbers, but intensities of land use that are clearly different (e.g. artificial meadow and broad-leafed forest). Negative impact values indicate that land use types hold more species per m² than the reference does. In terms of species diversity, these land use types are superior. In Table 5 $EDP^{S_{total}}_{linear}$, with a few number of plots – and therefore, high standard error – are flagged with #.

The nonlinear transformation resulting in $EDP^{S_{total}}_{nonlinear}$ is shown in **Table 6**. Since α-diversity of plants, moss and mollusks is uncorrelated or correlated with low R^2 (see Table 3), rankings of land use types are different according to whether they are based on $EDP^{S_{plants}}_{nonlinear}$, $EDP^{S_{moos}}_{nonlinear}$ or $EDP^{S_{mollusks}}_{nonlinear}$.

Although the type of transformation and species groups included differ, $EDP^{S_{total}}_{linear}$ and $EDP^{S_{total}}_{nonlinear}$ values for intensity classes are very similar. Remarkable differences have values for low intensity forests. This is explained by the fact that those forests have a rather low α-diversity of threatened plants and, because of their humid habitat conditions, a high α-diversity of moss and mollusks.

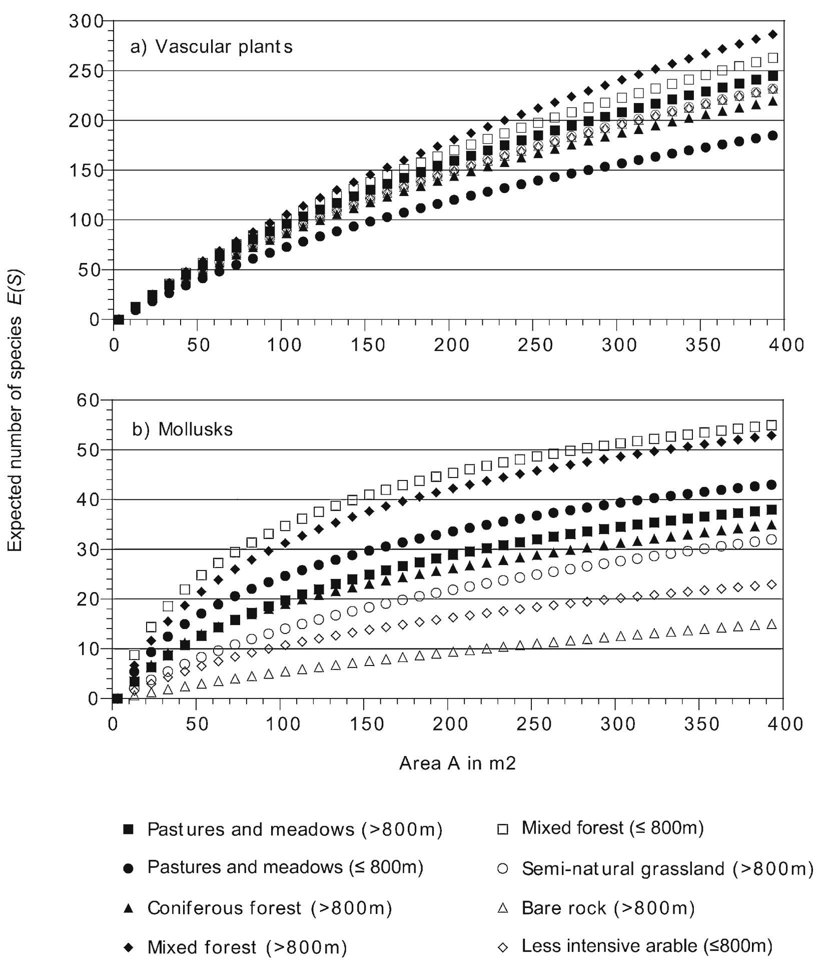

Fig. 3: Expected number of species based on rarefaction functions for vascular plants (a) and mollusks (b)

Table 4: Curve parameter for the power function $E(S)=cA^z$ fitted to a discontinuous rarefaction function. A distinction is made between land use types below 800 m above sea level and above

Land-use type	CORINE Plus ID	Plants		Mollusks	
		c	*z*	*c*	*z*
Less intensive arable (≤800m)	2112	2.39	0.78	0.44	0.68
Pastures and meadows (>800m)	231	2.49	0.78	0.90	0.61
Pastures and meadows (≤800m)	231	1.91	0.78	0.47	0.52
Coniferous forest (>800m)	312	2.21	0.78	0.71	0.55
Mixed forest (>800m)	313	2.37	0.81	0.36	0.51
Mixed forest (≤800m)	313	2.47	0.79	0.25	0.45
Semi-natural grassland (>800m)	321	2.12	0.80	0.48	0.72
Bare rock (>800m)	332	2.20	0.79	0.11	0.83

Table 5: Local ecosystem damage potential based on total plant species data and data on threatened plant species (EDP^{S}_{linear}). Means and standard error are calculated according to the linear function (equation [6]) with the Swiss Plateau as reference. EDP^{S}_{linear} values are dimensionless, but refer to 1 m² of specific land use types, intensity classes and the Swiss regions. Uncertain values of $EDP^{S_{total}}_{linear}$ with standard error above 0.2 are flagged with #. Those are not recommended for use in LCIA

		$EDP^{S_{plants}}_{linear}$					$EDP^{S_{threatened\_plants}}_{linear}$			$EDP^{S_{total}}_{linear}$		
CORINE Plus ID		Mean	Std. Error	Maximum	Minimum	N Plots	Mean	Std. Error	N Plots	Mean	Std. Error	
	Land use types											
111	Continuous urban	0.78	0.03	0.99	–0.68	65	–4.67	–	1	0.70	0.08	
112	Discontinuous urban	0.39	0.03	0.96	–1.04	152	–4.67	0.00	3	0.30	0.07	
113	Urban fallow	0.01	0.06	0.46	–0.85	27			0	0.01	0.06	
114	Rural settlement	0.38	0.03	0.65	0.22	17			0	0.38	0.03	
121	Industrial units	0.06	0.43	0.96	–1.36	5	–4.67	0.00	2	–1.80	1.51	#
121b	Industrial area with vegetation	0.39	0.05	0.95	–0.26	53			0	0.39	0.05	
122	Road and rail networks	–0.13	0.19	0.96	–1.36	18	–4.67	–	1	–0.39	0.34	#
122b	Road embankments	0.59	0.04	0.91	0.39	13			0	0.59	0.04	
122d	Rail embankments	0.10	0.04	0.44	–0.80	44			0	0.10	0.04	
122e	Rail fallow	0.41	0.03	0.63	0.14	19			0	0.41	0.03	
125	Industrial fallow	–0.01	0.05	0.50	–0.44	26			0	–0.01	0.05	
132	Dump sites	–0.08	0.25	0.28	–0.56	3			0	–0.08	0.25	#
134	Mining fallow	0.05	0.05	0.29	–0.24	10			0	0.05	0.05	
141	Green urban areas	0.26	0.04	0.87	–0.88	111	–4.67	–	1	0.22	0.06	
142	Sport and leisure facilities	0.75	0.07	0.92	–0.28	18			0	0.75	0.07	
211	Non-irrigated arable land	0.40	0.10	0.76	–0.36	12	–4.67	0.00	2	–0.38	0.61	#
211a	Intensive arable	0.74	0.02	0.98	0.38	54			0	0.74	0.02	
211b	Less intensive arable	0.75	0.02	0.99	–0.64	198	–5.80	1.13	5	0.61	0.08	
211c	Organic arable	0.36	0.03	0.94	0.03	62			0	0.36	0.03	
211d	Fibre/energy crops	0.68	0.02	0.99	0.27	94			0	0.68	0.02	
211e	Agricultural fallow	–0.08	0.03	0.71	–1.07	139	–4.67	–	1	–0.11	0.05	
211f	Artificial meadow	0.29	0.04	0.64	–0.04	28			0	0.29	0.04	
221	Vineyards	0.57	0.20	0.88	0.00	4			0	0.57	0.20	#
221b	Organic vineyards	0.42	0.03	0.68	–0.08	48			0	0.42	0.03	
222	Fruit trees and berry plantations	0.03	0.13	0.36	–0.20	4			0	0.03	0.13	#
222a	Intensive orchards	0.13	0.20	0.52	–0.12	3	–4.67	0.00	2	–2.98	1.75	#
222b	Organic orchards	0.10	0.34	0.44	–0.24	2			0	0.10	0.34	#

Table 5: Local ecosystem damage potential based on total plant species data and data on threatened plant species (EDP^{S}_{linear}). Means and standard error are calculated according to the linear function (equation [6]) with the Swiss Plateau as reference. EDP^{S}_{linear} values are dimensionless, but refer to 1 m² of specific land use types, intensity classes and the Swiss regions. Uncertain values of $EDP^{S_{total}}_{linear}$ with standard error above 0.2 are flagged with #. Those are not recommended for use in LCIA (*cont'd*)

		$EDP^{S_{plants}}_{linear}$					$EDP^{S_{threatened\_plants}}_{linear}$			$EDP^{S_{total}}_{linear}$		
CORINE Plus ID		**Mean**	**Std. Error**	**Maximum**	**Minimum**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	
231	Pastures and meadows	–0.01	0.04	0.60	–1.24	78			0	–0.01	0.04	
	" above 800m	–0.59	0.06	0.36	–2.00	86	–5.54	0.59	13	–1.42	0.26	#
231a	Intensive pasture and meadows	0.54	0.04	0.94	–0.92	73	–7.50	2.83	2	0.33	0.18	
231b	Less intensive pasture and meadows	0.52	0.03	0.90	–0.16	104			0	0.52	0.03	
231c	Organic pasture and meadows	–0.13	0.02	0.87	–1.84	727			0	–0.13	0.02	
244	Agro–forestry areas	–0.36	0.24	–0.12	–0.60	2			0	–0.36	0.24	#
245	Agricultural fallow with hedgerows	–0.32	0.05	–0.12	–0.62	11			0	–0.32	0.05	
311	Broad–leafed forest	0.31	0.06	0.92	–0.68	31	–4.67	0.00	2	0.01	0.22	#
	" above 800m	0.36	0.08	0.92	–0.64	26	–4.67	–	1	0.18	0.21	#
311a	Broad leafed plantations	0.49	0.13	0.75	0.00	5			0	0.49	0.13	#
311b	Semi–natural broad–leafed forests	0.41	0.01	0.94	–0.75	1312			0	0.41	0.01	
312	Coniferous forest	0.56	0.07	0.68	0.36	4			0	0.56	0.07	#
	" above 800m	0.15	0.05	0.88	–0.88	73	–6.09	0.93	8	–0.51	0.27	#
312a	Coniferous plantations	0.57	0.02	0.92	–0.16	74			0	0.57	0.02	
312b	Semi-natural coniferous forests	–0.03	0.05	0.76	–1.16	99			0	–0.03	0.05	
	" above 800m	–0.76	0.09	0.03	–1.09	13			0	–0.76	0.09	
313	Mixed forest	0.36	0.04	0.96	–0.82	83			0	0.36	0.04	
	" above 800m	–0.24	0.04	0.80	–1.34	223			0	–0.24	0.04	
313a	Mixed broad-leafed forest	0.19	0.07	0.44	–0.13	8			0	0.19	0.07	#
313b	Mixed coniferous forest	0.54	0.02	0.70	0.28	35			0	0.54	0.02	
313c	Mixed plantations	0.73	0.02	0.89	0.30	61			0	0.73	0.02	
314	Forest Edge	–0.19	0.07	0.94	–1.55	78			0	–0.19	0.07	
321	Semi-natural grassland	–0.18	0.03	0.89	–2.16	331	–5.26	0.41	19	–0.48	0.09	
322	Moors and heath land	0.09	0.07	0.70	–0.88	36	–7.50	2.83	2	–0.33	0.32	#
324	Transitional woodland/shrub	–0.11	0.07	0.90	–1.20	60	–4.67	0.00	2	–0.26	0.13	
331	Beaches, dunes, and sand plains	0.64	0.16	0.80	0.48	2			0	0.64	0.16	#
332	Bare rock	0.44	0.06	0.96	–0.44	42	–10.34		1	0.19	0.27	#
333	Sparsely vegetated areas	–0.27	0.09	0.68	–1.72	31	–4.67	0.00	6	–1.17	0.37	#
411	Inland marshes	–0.15	0.24	0.40	–0.80	5	–8.45	1.89	3	–5.22	2.13	#
412	Peat bogs	0.54	0.01	0.96	–0.54	634			0	0.54	0.01	
511	Water courses	0.52	0.17	0.88	0.00	5	–4.67	–	1	–0.41	1.07	#
	Total local plots	0.24	0.01	0.99	–2.16	5582	–5.54	0.23	78	0.17	0.01	
	Intensity classes											
1	Artificial surfaces	0.40	0.02	0.99	–1.36	481	–4.67	0.00	7	0.33	0.04	
2	Agriculture high intensity	0.63	0.01	0.99	–0.92	524	–5.70	0.69	11	0.51	0.04	
2	Agriculture low intensity	–0.06	0.02	0.90	–2.16	1214	–5.26	0.41	19	–0.15	0.03	
3	Forestry high intensity	0.63	0.02	0.92	–0.16	140			0	0.63	0.02	
3	Forestry low intensity	0.29	0.01	0.96	–1.34	1773			0	0.29	0.01	
3	Non-use	0.29	0.01	0.96	–1.72	1120	–6.18	0.67	15	0.21	0.03	
	Regions in Switzerland											
	Alps	–0.27	0.01	0.21	–1.13	202	–5.56	0.36	172	–5.00	0.35	
	Jura	–0.10	0.03	0.25	–0.49	44	–5.19	0.63	33	–3.99	0.59	
	Plateau	0.97	0.00	0.99	0.93	28	–0.02	0.17	23	0.96	0.14	
	Above timberline	0.00	0.02	0.30	–0.55	104	–7.67	0.59	87	–6.41	0.58	
	Lakes	0.31	0.01	0.78	–0.40	215	–0.37	0.07	177	0.01	0.06	

Table 6: Local ecosystem damage potential based on data on plant, moss and mollusk species ($EDP^{S}_{nonlinear}$). For their calculation, the non-linear model was used according to equation [8]. They are dimensionless, but refer to 1 m² of land. Uncertain values of $EDP^{S_{total}}_{nonlinear}$ with low number of sample plots n below 10 or standard error above 0.2 are flagged with #. Those are not recommended for use in LCIA. Like the linear EDP

		$EDP^{S_{plants}}_{nonlinear}$			$EDP^{S_{moos}}_{nonlinear}$			$EDP^{S_{mollusk}}_{nonlinear}$			$EDP^{S_{total}}_{nonlinear}$			
CORINE Plus ID		**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	**Mean**	**Std. Error**	**N Plots**	
	Land use type													
111	Continuous urban	0.24	0.10	5	0.42	0.06	5	–0.10	0.19	3	0.59	0.18	5	#
112	Discontinuous urban	0.10	0.05	25	0.25	0.04	23	–0.01	0.04	20	0.32	0.08	25	
121	Industrial units	0.14	0.25	4	0.32	0.10	4	0.13	0.16	3	0.56	0.34	4	#
122	Road and rail networks	0.08	0.10	13	0.22	0.06	12	–0.10	0.07	11	0.20	0.16	13	
132	Dump sites	–0.01	0.06	3	0.30	0.06	3	0.35	0.10	3	0.65	0.19	3	#
141	Green urban areas	–0.17	–	1	0.08	–	1	–0.17	–	1	–0.27	–	1	#
142	Sport and leisure facilities	0.07	0.13	2	0.44	0.28	2	–0.33	–	1	0.34	0.58	2	#
211	Non–irrigated arable land	0.22	0.04	8	0.48	0.07	5	0.23	0.07	5	0.66	0.12	8	#
211b	Less intensive arable	0.20	0.03	30	0.61	0.04	15	0.14	0.04	24	0.62	0.07	30	
211e	Agricultural fallow	–0.14		1	0.20	–	1	0.15	–	1	0.21	–	1	#
211f	Artificial meadow	0.10	0.01	26	0.48	0.03	22	0.06	0.04	23	0.56	0.06	26	
221	Vineyards	0.31	0.12	4	0.72		1	–0.06	0.09	4	0.43	0.15	4	#
222	Fruit trees and berry plantations	0.02	0.04	4	0.52	0.15	4	–0.07	0.03	4	0.47	0.09	4	#
222a	Intensive orchards	0.06	0.07	3	0.44	0.09	2	0.00	0.09	3	0.35	0.20	3	#
222b	Organic orchards	0.05	0.11	2	0.72	–	1	–0.05	0.12	2	0.36	0.59	2	#
231	Pastures and meadows	0.02	0.01	74	0.48	0.03	60	0.03	0.03	72	0.43	0.04	74	
	" above 800m	–0.11	0.01	73	0.22	0.03	69	0.08	0.03	70	0.18	0.05	73	
231a	Intensive pasture and meadows	–0.10	0.07	3	0.36	0.06	3	0.04	0.12	3	0.31	0.13	3	#
244	Agro-forestry areas	–0.08	0.05	2	–0.01	0.14	2	0.08	0.07	2	0.00	0.26	2	#
311	Broad-leafed forest	0.13	0.03	30	0.09	0.03	30	–0.18	0.04	29	0.06	0.06	30	
	" above 800m	0.19	0.04	25	0.11	0.04	25	0.03	0.06	21	0.33	0.10	25	
311b	Semi-natural broad-leafed forests	0.04	0.01	115	0.61	0.02	115	–	–	0	0.65	0.02	115	
312	Coniferous forest	0.27	0.03	3	0.00	0.08	3	–0.13	0.11	3	0.14	0.16	3	#
	" above 800m	0.09	0.02	73	0.00	0.02	73	0.06	0.03	66	0.15	0.04	73	
312a	Coniferous plantations	0.22	0.10	6	0.14	0.04	6	–0.04	0.03	6	0.31	0.15	6	#
312b	Semi-natural coniferous forests	0.09	0.30	2	0.00	0.01	2	–0.09	0.05	2	0.00	0.23	2	#
313	Mixed forest	0.19	0.03	36	0.15	0.02	35	–0.16	0.03	36	0.17	0.05	36	
	" above 800m	0.12	0.02	47	0.03	0.02	46	–0.07	0.04	46	0.09	0.06	47	
313b	Mixed coniferous forest	0.15	0.05	2	0.08	0.05	2	0.02	0.00	2	0.25	0.00	2	#
321	Semi-natural grassland	–0.13	0.01	88	0.05	0.02	86	0.09	0.03	66	–0.01	0.03	88	
322	Moors and heath land	0.02	0.02	30	–0.03	0.02	30	0.11	0.06	21	0.07	0.06	30	
324	Transitional woodland/shrub	–0.01	0.03	16	0.02	0.05	16	–0.02	0.06	13	0.00	0.08	16	
331	Beaches, dunes, and sand plains	0.31	0.13	2	0.54	0.19	2	0.15	–	1	0.92	0.39	2	#
332	Bare rock	0.23	0.03	41	0.13	0.03	40	0.23	0.05	18	0.45	0.06	41	
333	Sparsely vegetated areas	–0.04	0.02	31	–0.02	0.03	30	0.19	0.05	23	0.08	0.06	31	
411	Inland marshes	–0.01	0.06	5	0.29	0.10	5	–0.23	0.10	5	0.05	0.13	5	#
412	Peat bogs	0.05	–	1	0.29	–	1	–0.08	–	1	0.26	–	1	#
511	Water courses	0.28	0.11	5	0.20	0.07	5	0.24	0.13	3	0.63	0.16	5	#
	Intensity class													
1	Artificial surfaces	0.11	0.04	40	0.29	0.03	38	0.01	0.04	31	0.39	0.07	40	
2	Agriculture high intensity	0.15	0.02	70	0.51	0.02	47	0.11	0.03	58	0.58	0.04	70	
2	Agriculture low intensity	–0.13	0.01	92	0.06	0.02	89	0.09	0.03	70	–0.01	0.03	92	
3	Forestry high intensity	0.22	0.10	6	0.14	0.04	6	–0.04	0.03	6	0.31	0.15	6	#
3	Forestry low intensity	0.09	0.01	202	0.39	0.02	200	–0.11	0.02	86	0.43	0.03	202	
3	Non-use	0.07	0.02	127	0.06	0.02	125	0.12	0.03	83	0.20	0.04	127	

2.5 Use of characterization factors EDP^S for calculation of damages for land occupation and land use change

In this section, we explain how to use the characterization factors EDP^S in Life Cycle Impact Assessment. For calculation of the *damage of land occupation*, we refer to equation (2) of Part 1 of Koellner and Scholz (2007). For a specific land use type the time of occupation in years is multiplied with the area in m² and also multiplied with the EDP factor for this specific land use type. We recommend using linear $EDP^{S_{total}}_{linear}$ from Table 5. They are more robust compared to the nonlinear version.

More difficult is the situation when calculating the *damage of land use change*. We argued in Section 3.3 of Part 1 that land use change includes damage from transformation and restoration. The damage for transformation and restoration are calculated according to equations (5) and (6) (Koellner and Scholz 2007). Taking, for example, the land use change from low intensity forest into high intensity agriculture, the damage is without taking the baseline damage into account:

$$\begin{aligned} D_{change} &= D_{trans} + D_{rest} = \\ & A_{trans} \cdot T_{trans} \cdot \left[\frac{1}{2} \left| \left(EDP(\cdot, t_1) - EDP(\cdot, t_0) \right) \right| \right] \\ & + A_{rest} \cdot T_{rest} \cdot \left[\frac{1}{2} \left| \left(EDP(\cdot, t_3) - EDP(\cdot, t_2) \right) \right| \right] = \\ & A_{trans} \cdot T_{forest\_li->agri\_hi} \cdot \left[\frac{1}{2} \left| \left(EDP_{agri\_hi} - EDP_{forest\_li} \right) \right| \right] \\ & + A_{rest} \cdot T_{agri\_hi->forest\_li} \cdot \left[\frac{1}{2} \left| \left(EDP_{forest\_li} - EDP_{agri\_hi} \right) \right| \right] \end{aligned} \tag{9}$$

Assuming that $A_{change} = A_{trans} = A_{rest}$ and with $EDP_{agri\_hi} = 0.51$ and $EDP_{forest\_li} = 0.29$ from Table 5, and with $T_{forest\_li->agri\_hi} = 1$, $T_{agri\_hi->forest\_li} = 50$ from Table 2 (Part 1 of this paper series) it is

$$\begin{aligned} D_{change} &= A_{trans} \left(1 \cdot \left[\frac{1}{2} \cdot \left| \left(0.51 - 0.29 \right) \right| \right] + 50 \cdot \left[\frac{1}{2} \cdot \left| \left(0.29 - 0.51 \right) \right| \right] \right) = \\ & A_{trans} \left(1 \cdot \frac{1}{2} \cdot 0.22 + 50 \cdot \frac{1}{2} \cdot 0.22 \right) = A_{trans} \cdot \left(0.11 + 5.5 \right) = A_{trans} \cdot 5.61 \end{aligned} \tag{10}$$

In this example, the phase of transformation from forest into agriculture accounts for less damage than the phase of the restoration of the forest. The main reason for the difference is the long restoration time in relation to transformation time. For practical applications, the damage of the transformation phase could be neglected, if transformation is rapid compared to restoration. The damage of changing 1 unit area from low intensity forest to high intensity agriculture is more then ten times higher compared to the occupation of 1 unit area of existing high intensity agriculture for 1 year ($D_{occ} = A_{occ} \cdot T_{occ} \cdot EDP_{agri\_hi} = A_{occ} \cdot 1 \cdot 0.51$). This example shows that it makes sense to maximize occupation time of existing high intensity agriculture and minimize transformation of forest into agricultural land use to achieve constant functional output.

3 Discussion

3.1 Validity of ecosystem damage potential EDP^S

Land use has severe impacts, not only on biodiversity, but also on ecosystem services (e.g. water purification, carbon sequestration, biomass productivity) and scenic beauty (Daily 1997). A comprehensive assessment of the ecosystem damage potential of land use requires the integration of all those aspects. We focused on biodiversity, because it is an important aspect of the ecosystem. Biodiversity is regarded as a key element for ecosystem functioning (Naeem and Li 1997, Schläpfer and Schmid 1999, Schulze and Mooney 1994) and its intrinsic value is stressed by the Convention on Biological Diversity (UNEP 1992).

The ecosystem damage potential EDP^S is based on assessment of impacts of land use on species diversity. We clearly base EDP^S factors on α-diversity, which correlates with the local aspect of species diversity of land use types. Based on an extensive meta-analysis of biologists' field research, we were able to include data on the diversity of plant species, threatened plant species, moss and mollusks in the EDP^S. The integration of other animal species groups (e.g. insects, birds, mammals, amphibians) with their specific habitat preferences could change the characterization factors values specific for each land use type. Ecosystem functions are supported by those mobile species groups, because they provide functional links between habitats in the landscape (Lundberg and Moberg 2003). Many studies propose characterization factors based on α-diversity. More specifically these focus either on species lost or absolute number of species on the local scale. Factors proposed are potentially disappeared fraction of vascular plant species (Goedkoop et al. 1998, Goedkoop and Spriensma 1999) or loss of vascular plant species per area (Udo de Haes et al. 1999), Other propose to take the number of species separated into different groups, i.e. tree, shrub, and herb species (Schweinle 1998), diversity of trees and structural diversity of forest (Giegrich and Sturm 1996) or number of rare species and number of all species (Cowell 1998). We propose also a relative indicator for α-diversity, because absolute numbers of local species diversity generally increase from South to North, even for the same type of land use type.

In addition β-diversity is an important aspect of species diversity, because not only absolute or relative species numbers, but similarities of species composition between different plots of one land use types are compared. Lindeijer et al. (1998) proposed to take species accumulation rate per land use type as an indicator. Changes are assessed on a cardinal scale for different ecosystems worldwide. Koellner (2004) gives examples of an indicator for β-diversity for different functional species groups, which are somehow, linked to different land use types (e.g., forest species, unfertilized meadow species, fertilized meadow species). There was a large difference found between the functional species groups, however, data are not sufficient to derive characterization factors for LCA, because the link to specific land use types is not clear enough. In contrast the factors calculated in this paper (see Table 4) are directly linked to specific land use types, but results are ambiguous, mainly because of data

limitations, which are expected to less severe in the future. For this reasons β-diversity could not be included in the calculation of EDP^S in this paper,

Vogtländer (2004) proposed characterization factors based on the diversity of ecosystems and Müller-Wenk (1998) on the occurrence of rare ecosystems on the landscape scale. In our assessment, this has been integrated indirectly, because we took the number of threatened plant species into account. Generally speaking, species which are bound to scarce ecosystem types are rare on the landscape scale. Semi-natural grassland, moors, heath land and inland marshes are such rare ecosystem types and show high number of threatened species. These ecosystem types have severely lost areas in the course of the development of industrial agriculture, intensive forestry and urban areas sprawl.

Such characterization factors refer to γ-diversity and to the assessment of regional impacts of land use. A proposal how to assess regional land use impacts based on a regression analysis with species potentially lost between about 1850 and 1975 (dependent variable) and land use on the regional scale (independent variable) was given by Koellner (2003). Although the quality of the available data was high, the results were not very robust and limited to the specific regional characteristics of Switzerland. But again, with better data availability in the future, it will be possible to calculate characterization factors based also on this aspect of biodiversity.

The *linear transformation* of species data into $EDP^{S_{total}}_{linear}$ assigns the same value to each plant species and allows for accounting the non-use value of species diversity. For the same reason, we also integrated the number of threatened species separately. Since data on moss and mollusks diversity was only available for a sub-sample (841 out of 5,581 plots), it was not included here. The *nonlinear transformation* into $EDP^{S_{total}}_{nonlinear}$ was developed to account for the functional aspect of species diversity. According to the redundant species hypothesis, the relationship between species loss and ecosystem functioning is nonlinear. The reasoning behind this hypothesis is that species are redundant, as demonstrated by the fact that a specific ecological function (e.g. fixation of nitrogen) can be fulfilled by different species (Lawton 1996). The assumption underlying this is that all of the species within one functional group are equally adapted for fulfilling the same specific function (Schulze and Mooney 1994, pp. 501), but in fact, species within one functional group differ, particularly in their response to environmental changes. Formerly, redundant species might become important for ecosystem functioning, because they are better adapted to new environmental conditions. This point is very much stressed in the rivet hypothesis, framed by Ehrlich and Ehrlich (1981).

The empirical data used for the development of EDP^S were acquired from Switzerland and Germany. The external validity refers to the question of whether EDP^S can be generalized to other regions or countries. Most of the data was sampled from regions which were subject to intensive use and consist largely of agriculture, forestry, and urban land. One can expect that the absolute species number will be valid for regions which are similar in land use intensity and biogeographical situation. We used a relative measure for local species richness and took the regional species richness as a reference. As a result, the error might be less significant when the findings are generalized to other European countries. The coarse ranking of the land use types is expected to be stable across a wide geographical range. The calculated EDP^S are expected to be valid for the European part of diversity zones 5 and 6 of Barthlott's (1999) map of global diversity.

Limitation of the approach is that a land use is only specified in terms of type, area and duration. Specific characteristics of patch size and shape, location of patches in the landscape and their fragmentation have large impacts on abundance and diversity of species (Fahrig and Jonsen 1998). Due to data limitations, all those factors could not be included in this assessment.

One issue pertaining to *internal validity* is whether calculated factors only refer to land use impacts or also refer to other impacts. In the latter case, double counting of impacts would occur, since some other impact categories (e.g. nutrification and ecotoxicological effects) are assessed separately in LCIA. In contrast to the suggestions of Udo de Haes (2006) and Milà i Canals et al. (2007), the characterization factors for land use calculated in this paper integrate all the impacts which result from land use. These impacts include the intentional application of chemicals, such as in agriculture, where fertilizer and pesticides are used, as well as physical impacts like ploughing. If one wants to include damage to biodiversity in LCIA, it is not possible to separate those chemical and physical impacts. However, it is necessary to distinguish (i) impacts on the plot of land use from (ii) impacts outside of this immediate plot resulting from runoff. In the former case, both the physical and chemical impacts of land use are included in the characterization factor EDP^S. However, in the latter case, damage due to impacts from runoff are not included in the EDP^S factor, but must be considered in other impact categories. One can conclude from this that double counting is not a problem, when local EDP^S factors are used, which refer only to the plot in use. However, if regional factors referring to γ-diversity could be calculated, double counting would be a problem to be addressed.

3.2 Uncertainty of characterization factors *EDP^S*

In EDP^S, the uncertainty of the results is strongly influenced by the empirical data basis of the species diversity. An important source of uncertainty in meta-analysis is the reliability. This refers to the stability of results when many researchers have conducted the investigations or different methods have been used. In our meta-analysis on species diversity, we took 23 different sources into account. The data for species diversity was mostly sampled according to the standardized method from Braun-Blanquet, which is applied in vegetation science, although other methods were also used to determine species richness. The accuracy of data can also vary from researcher to researcher, but it is difficult to judge this aspect of reliability on the basis of the available literature. The data on plant moss and mollusk species for the 841 plots from the Biodiversity Monitoring Switzerland are highly reliable.

The number of plots investigated has also a strong influence on the reliability of EDP^S. The uncertainty for α-diversity of plant species is generally rather low as a result of the high sample numbers. Uncertainties were measured with the standard error accounting for the standard deviation as well as the number of plots sampled. The uncertainty of α-diversity of moss and mollusks is higher because of their smaller sample sizes. The number of plots investigated varies across land use types, resulting in large differences in uncertainty. An important source of uncertainty is the limited data availability for threatened species. For many land use types, no data were available for this aspect of species diversity (see Table 5). Nevertheless, we found that the inclusion of data on threatened species results in correction of those land use types with a high value for biodiversity (e.g. moors and heathland). Critical *EDP* values with low *n* and large standard error are flagged (#) and should not be used in LCA.

The standardization of species numbers is very important for reliable EDP^S. The sampling method has a strong influence on the size and number of the plots investigated. In general, with Braun-Blanquet, many small plots are sampled; with the other method, few large plots are investigated. The species-area curve allows a reliable standardization of plant species numbers across every relevant scale from 1m² to 10,000 km² (see Fig. 2). As a reliability check, we standardized the values of Barthlott's (1999) map of global diversity, which were not included in the regression function. For diversity zone 5, we obtained a lower margin of 8.9 S/m² and an upper margin of 13.3 S/m², which is rather close to the empirical value of 15.6 S/m² of the Swiss Plateau. For diversity zone 6, margins (13.3–17.7 S/m²) were also very close to the value of the Swiss Alps (19.7 S/m²). Standardization for other diversity zones and species groups needs to be further investigated.

A first attempt to integrate β-diversity into the impact assessment was undertaken. This aspect of diversity facilitates a regional assessment, because species turnover can be assessed across the landscape. It can be used to indicate functional opportunities of species diversity in a given landscape and, thus, relate to ecosystem resilience (Peterson et al. 1998). We applied the rarefaction method proposed by Hurlbert (1971) and Heck (1975) for quantification of β-diversity for vascular plants and mollusks. As proposed by Ricotta (2002), the slope *z* of the fitted rarefaction curve was taken as an indicator for β-diversity. The *z*-values, however, did not provide a clear differentiation of land use types. This is most presumably because rarefaction curves based on a small total area (39 plots per land use type with a size of 10 m² each) did not come to saturation. The low reliability of these results prevented their use in calculating characterization factors. In cases where the species composition of larger sample areas is known, however, this method has proved useful for calculating β-diversity on the regional scale (Koellner et al. 2004) and could be used to derive characterization factors, if sufficient data would be available.

For practical applications, we recommend using the EDP^S_{linear} linear values. The values for $EDP^S_{nonlinear}$ nonlinear are calculated from only one data set (Biodiversity Monitoring Switzerland). Therefore, the uncertainty is large because many land use types were available for which data were only available for a few number of plots.

4 Conclusions

An impact assessment method for land use with generic characterization factors (EDP^S) improves the basis for decision-making in industry and other organizations. The bio-geographical differentiation of generic characterization factors is mandatory. We have suggested that Barthlott's ten diversity zones can be used as a basis to develop a LCIA method, which can be used in a global context. The challenge, however, will be to find sufficient empirical information on species diversity to cover all diversity zones with characterization factors for all relevant land use types. The method developed here can best be applied to marginal land use decisions; that is, to decisions in which the consequences are so small that the quality or quantity of environmental parameters of a region is not noticeably altered. However, many of these marginal decisions on a micro level can have a substantial impact on the environment. We focused on this type of application, because LCA is a tool for supporting decisions on a micro level. In order to support decisions on a macro level (e.g. policy decisions restricting intensive agriculture) a non-marginal approach is advisable and the method developed here must be completed with a regional assessment (in contrast to the local assessment here, a regional assessment would address the expected changes of biodiversity in a region due to land use, see Koellner 2003). In order to support decisions on distinct land use projects involving a generic assessment, these should be accomplished with site-dependent assessment methods.

Acknowlegement. For providing data on species diversity for different land use types, we would like to thank Urs Hintermann (Biodiversity Monitoring Switzerland), Erich Kohli (BUWAL, Bern), Thomas Wohlgemuth and Thomas Dalang (WSL, Birmensdorf), and Andrea Schwab and Andrea Lips (FAL, Reckenholz). We would also like to thank Ruedi Müller-Wenk for valuable discussion and comments on an earlier version of this paper and two reviewers for their important feedback.

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Received: September 1st, 2005
Accepted: December 8th, 2006
OnlineFirst: December 9th, 2006

Int J LCA 12 (1) 16–23 (2007)

Assessment of Land Use Impacts on the Natural Environment

Part 1: An Analytical Framework for Pure Land Occupation and Land Use Change

Thomas Koellner* and Roland W. Scholz

Swiss Federal Institute of Technology, Department of Environmental Sciences, Natural and Social Science Interface (ETH-NSSI), ETH-Zentrum HAD, 8092 Zurich, Switzerland

* **Corresponding author** (thomas.koellner@env.ethz.ch)

DOI: http://dx.doi.org/10.1065/lca2006.12.292.1

Abstract

Goal, Scope and Background. In the framework of LCA, land use is broadly accepted as an impact category. However, the methodology for the assessment of damages on the natural environment was and still is the subject of discussion. The main objective of this paper is to contribute to that discussion by providing a consistent methodological framework for the assessment of land occupation and transformation.

Methods. We clarify the context of LCA relevant land use decisions. Based on that, we develop a formal model with damage functions and generic characterization factors for quantifying damages on ecosystems from land occupation and land transformation. The characterization factor for land occupation and land use change is labeled Ecosystem Damage Potential (*EDP*). We analytically address the substitutability of area and time occupied in order to produce a constant output.

Results. Based on the proposed method, it is possible to calculate the damages from complex series of land transformation, land occupation, and land restoration. A main feature of the method is that land transformation is assessed based on a factual or virtual, restoration time. This means that the damage of land transformation is largest for land use types which are difficult to restore and need extremely long to develop (e.g. thousand of years and more for primary forest and peatbog). In addition, we could show that area and time of occupation are not substitutable. The more severe the damage potential of a specific land use type is, the better it is to minimize the area and maximize the duration of occupation.

Discussion. An approach for the assessment of pure land occupation and land use change was developed in this paper, which is not geographically referenced. Developing geographically-referenced land use inventories and impact assessment methods can increase their accuracy. The information cost to provide geographically referenced data on land use for practical LCA applications, however, would increase enormously.

Conclusions. An impact assessment method for land use with generic characterization factors improves the basis for decision-making in industry and other organizations. It can best be applied to marginal land use decisions; that is, to decisions in which the consequences are so small that the quality or quantity of environmental parameters of a region is not noticeably altered.

Recommendations and Perspectives. One main problem to address is the development of reliable generic characterization factors, which express the ecosystem damage potential of specific land use types. The characterization factors should be developed on an empirical basis, which allow decision makers to get access to knowledge from environmental sciences in a very condensed form. In order to support decisions on distinct land use projects, methods should be developed, which allow accomplishing a generic assessment with site-dependent assessments.

Keywords: Characterization factor; damage function; ecosystem; impact; land use; LCA; restoration time